Science of the Total Environment 621 (2018) 253–264

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Quantifying the combined effects of land use and climate changes on stream flow and nutrient loads: A modelling approach in the Odense Fjord catchment (Denmark)

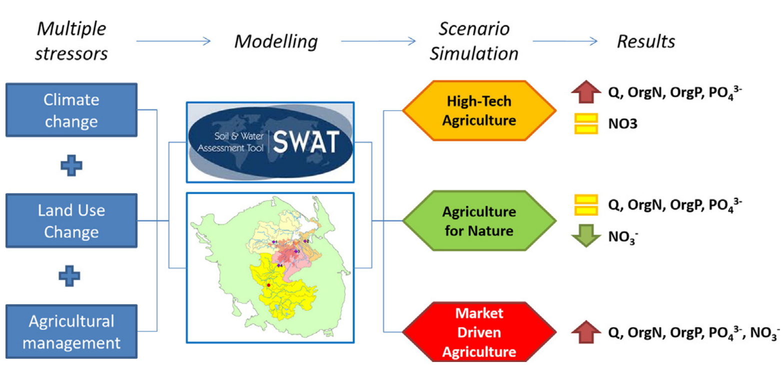
Eugenio Molina-Navarro *, Hans E. Andersen, Anders Nielsen, Hans Thodsen, Dennis Trolle

Department of Bioscience, Aarhus University, Silkeborg, Denmark

HIGHLIGHTS

- We calibrated a SWAT model for discharge and four nutrients in the Odense catchment.
- We downscaled three future storylines focusing on climate and land use changes.
- Changes in land use alone showed an impact on NO_3^- loss due to changes in fertilization.
- Discharge, organic nutrients and PO_4^{3-} loads will increase under high emissions.
- Both land use and climate changes will interact regarding nitrate loads.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 19 May 2017
Received in revised form 7 November 2017
Accepted 21 November 2017
Available online 1 December 2017

Keywords:
Catchment management
Climate change
Eco-hydrological modelling
Nutrients
Scenario simulation
SWAT

ABSTRACT

Water pollution and water scarcity are among the main environmental challenges faced by the European Union, and multiple stressors compromise the integrity of water resources and ecosystems. Particularly in lowland areas of northern Europe, high population density, flood protection and, especially, intensive agriculture, are important drivers of water quality degradation. In addition, future climate and land use changes may interact, with uncertain consequences for water resources. Modelling approaches have become essential to address water issues and to evaluate ecosystem management. In this work, three multi-stressor future storylines combining climatic and socio-economic changes, defined at European level, have been downscaled for the Odense Fjord catchment (Denmark), giving three scenarios: High-Tech agriculture (HT), Agriculture for Nature (AN) and Market-Driven agriculture (MD). The impacts of these scenarios on water discharge and inorganic and organic nutrient loads to the streams have been simulated using the Soil and Water Assessment Tool (SWAT). The results revealed that the scenario-specific climate inputs were most important when simulating hydrology, increasing river discharge in the HT and MD scenarios (which followed the high emission 8.5 representative concentration pathway, RCP), while remaining stable in the AN scenario (RCP 4.5). Moreover, discharge was the main driver of changes in organic nutrients and inorganic phosphorus loads that consequently increased in a high emission scenario. Nevertheless, both land use (via inputs of fertilizer) and climate changes affected the nitrate transport. Different levels of fertilization yielded a decrease in the nitrate load in AN and an increase in MD. In HT, however, nitrate losses remained stable because the fertilization decrease was counteracted by a flow increase. Thus, our results suggest that N loads will ultimately depend on future land use and management in an interaction with

* Corresponding author at: Department of Bioscience, Aarhus University, Vejlsøvej 25, 8600 Silkeborg, Denmark.
E-mail addresses: emna@bios.au.dk, eugenio.molinanavarro@gmail.com (E. Molina-Navarro).

https://doi.org/10.1016/j.scitotenv.2017.11.251

climate changes, and this knowledge is of utmost importance for the achievement of European environmental policy goals.

1. Introduction

Water pollution and water scarcity are among the main challenges faced by the European Union. The continuing presence of a range of pollutants (e.g., creating excessive nutrient levels) and the possible consequences of climate change constitute a threat to water resources and their associated ecosystems (Kristensen, 2012). The main aim of the EU water policy, anchored in the European Water Framework Directive (WFD), is to ensure that a sufficient quantity of good quality water is available for people's needs and for the environment. The Directive committed the EU member states to achieve good ecological and chemical status of all surface water bodies by 2015, which involved fulfilling certain quality and quantity standards (European Parliament and Council, 2000). However, according to Europe's first generation of River Basin Management Plans, many water bodies failed to achieve this target. Thus, the EU has planned a revision of the WFD by 2019 and set the new deadline for achieving the overall WFD targets by 2027 (Hering et al., 2015).

In northern Europe, multiple stressors compromise the integrity of water resources and ecosystems. In lowland areas, high population density, intensive agriculture and flood protection have been important drivers of water quality degradation, and freshwater streams and lakes, as well as transitional (estuaries) and coastal waters are severely affected by eutrophication and pollution (Hering et al., 2015). Eutrophication is a recurrent issue in the Baltic Sea and the affected countries have therefore adopted various counteracting measures, like the Baltic Sea Action Plan (HELCOM, 2007). In Denmark, one of the world's most intensively farmed countries, diffuse nutrient losses from agriculture to water bodies, especially nitrogen, are of particularly great concern (Kaspersen et al., 2016). As a result of the undertaking of Danish water action plans, the total contents of nitrogen and phosphorus in the streams have decreased by approximately 43% and 40%, respectively, since 1989 (Thodsen et al., 2016). However, further reductions are required to ensure successful implementation of the WFD.

Odense Fjord, located on the island of Funen (Denmark), is one of the ecosystems affected by environmental pollution. Thus, elevated nutrient input levels have led to hypoxia, algal blooms and disappearance of seabed vegetation and fauna (Conley et al., 2007; Miljøministeriet, 2011). The fjord has a catchment area of approx. 1100 km^2, including rivers and lakes. The integrity of water ecosystems in the area has been damaged due to urbanisation and use of fertilizers and pesticides derived from the industrialisation of the agricultural sector, extensive channelisation of rivers, summer droughts and groundwater abstraction (Miljø- og Fødevareministeriet, 2016a). For Funen as a whole, Henriksen et al. (2008) reported an abstraction slightly over the sustainable yield, with the highest over-exploitation taking place in the Odense Fjord catchment, probably due to abstraction for water supply of Funen's largest city, Odense.

Climate and land use changes may interact and thus have combined effects on water resources, and the consequences of this are uncertain. To take into account the effects of climate change on the availability of freshwater is one of the main challenges to water administrators in Europe (Kristensen, 2012). Despite this, climate change was not consistently considered in the first generation of European River Basin Management Plans. The latest assessment elaborated by the Intergovernmental Panel on Climate Change (IPCC) has reported a general warming trend all over Europe, with the strongest warming projected for Southern Europe in summer and Northern Europe in winter (Kovats et al., 2014). For precipitation, climate projections show a general increase in Northern Europe, mainly during winter, while precipitation may decrease in summer with the associated risk of drought due to enhanced evapotranspiration. Consequently, climate change is projected to affect the hydrology and conditions of agricultural production in the Nordic-Baltic region (Jensen and Veihe, 2009; Øygarden et al., 2014; Trolle et al., 2015). In lowland catchments, most studies predicted an increase in average flow, generally due to the higher precipitation during winter, including also an increase of extreme events (e.g. Karlsson et al., 2015; e.g. Thodsen, 2007; van Roosmalen et al., 2009). The occurrence of high flows may, however, decrease in spring due to reduced snowpack and earlier melt as a result of the higher temperatures (Arheimer and Lindström, 2015). Increased temperatures will prolong the length of the growing season, potentially up to 2–4 months in some Nordic locations, and milder winters will allow more productive cultivation of winter crops (Jensen and Veihe, 2009; Øygarden et al., 2014).

The changes in precipitation and runoff, as well as temperature increases, will expectedly lead to an increase in nutrient loadings to surface water bodies in Northern Europe. Several studies have predicted profound effects of climate change on nutrient loading and eutrophication (e.g. Andersen et al., 2006; Jeppesen et al., 2009; Trolle et al., 2011), but better understanding of how nutrient losses will be influenced by the climate-induced changes in hydrology and agricultural management practices is essential. For the Nordic-Baltic region, higher temperatures influencing nutrient mineralisation, and increased precipitation will expectedly result in enhanced nutrient losses outside the growing season (Øygarden et al., 2014). Predictions for the growing season are more uncertain since climate change effects may influence agricultural decisions about land use and management (Jensen and Veihe, 2009), and many factors (duration of the growing season, weather conditions, changes in fertilization and yields, mineralisation, plant survival, future policies) may affect nutrient losses (Øygarden et al., 2014). Trolle et al. (2015) suggested that it will not be possible to obtain 'good' ecological status required by the EU WFD without interfering substantially with land use and crop production.

Considering the multiple stressors on water resources, a holistic and multidisciplinary approach is needed to understand how these may interact and explore what might be done to mitigate the undesirable effects. For this purpose, simulation models have proven to be useful tools (Devia et al., 2015; Trolle et al., 2012). In particular, catchment-scale models are valuable in addressing water quantity and quality issues, to evaluate ecosystem management practices and ultimately to develop river basin management plans (Arnold et al., 1998; Schoumans et al., 2009). One of those models is the Soil and Water Assessment Tool (SWAT, Arnold et al., 1998), a physically-based eco-hydrological model developed by the US Department of Agriculture (USDA) and the Texas A & M University. SWAT is used worldwide and >2500 papers in international journals have been published based on its application and ongoing development.

The SWAT model has already been set up for the Odense catchment (Thodsen et al., 2015) and may thus be used as a platform to address water issues in the area. Some studies have also endeavoured to assess the impact of future climate scenarios in the Odense catchment. For example, Thodsen (2007) studied the influence of the IPCC A2 scenario on stream flow (2071–2010 vs. 1961–1990), and Karlsson et al. (2015) and Trolle et al. (2015) assessed the impacts of an extreme 6 °C warming scenario on future runoff and nutrient exports.

In this study, three storylines, combining climate and land use change (socio-economic) scenarios, were developed to help differentiate the effects of multiple stressors on water quality and availability. The main objective was to run multi-stressor scenarios with the SWAT

model to assess the impacts on water discharge and inorganic and organic nutrient loads to streams within the Odense Fjord catchment. Very detailed inputs (e.g. daily climate projections for five variables, detailed land use and agricultural management operations) were used to generate realistic future scenarios. To the best of our knowledge, this is the first time that the impacts of combined climate and land use change on both water quality and quantity are assessed at such a detailed level at catchment scale in Denmark. This work was carried out within the framework of the EU-funded MARS project (Managing Aquatic Ecosystems and Water Resources under Multiple Stress, www.mars-project.eu, Hering et al., 2015), in which future storylines for climate and land uses in Europe have been assessed.

2. Material and methods

2.1. Study area

The Odense Fjord catchment (latitude 55.32° N, longitude 10.36° E) is the major catchment in the Danish island of Funen (Fig. 1). It has an area of 1061 km^2 and includes the catchment of Odense River, which is the largest river draining into the Odense Fjord (which actually is an estuary). According to Köppen-Geiger Classification, the climate is oceanic (warm temperate, fully humid, Kottek et al., 2006). For the period 2000–2010, the annual mean temperature was 8.7 °C with July being the warmest month (17 °C average) and January the coldest (1 °C average). Mean annual precipitation was 812 mm, showing no pronounced seasonality.

The altitude ranges from −8 to 125 m a.s.l. and half of the area has surface slopes below 2%. The catchment is settled over clayey moraines from the last glaciation (Weichsel glaciation) (Smed, 1982) and sandy-loam soils dominate the catchment.

Land use is dominated by agriculture (68%), followed by urban areas (16%) and forest (10%). The dominant crops are winter wheat (42%), spring barley (21%) and oil seed rape (14%) (Thodsen et al., 2015). The catchment has undergone substantial hydrological and hydromorphological modifications, including sub-surface tile drainage in about half of the agricultural area (Thodsen et al., 2015). Odense, with 187,000 inhabitants, is the main city in the catchment. Many of the water bodies, including the estuary into which the catchment discharges, do not meet the WFD criteria of good ecological status (Miljø- og Fødevareministeriet, 2016b). Of the 600 km streams targeted by the WFD, only 36% have good or high ecological status, and the corresponding number for the 17 lakes larger than 5 ha is 12% (Miljø- og Fødevareministeriet, 2016b). The ecological status of the Odense Fjord estuary is conditioned by nutrient loads from the catchment. In spite of several national action plans, the ecological status of the estuary is classified as moderate/bad with a high risk of not achieving the environmental objective of good ecological status set in the next river basin management planning cycle (Miljø- og Fødevareministeriet, 2016b).

2.2. Model set-up, calibration and validation

The Odense River catchment model was set up by Thodsen et al. (2015) where a detailed description can be found. In short, the catchment was divided into 31 sub-basins (Fig. 1) and 11 soil types, 6 land uses and 3 slope classes. However, since agriculture is the major land use, it is highly important to represent it properly in the model to account fully for nutrient exports. Following the regional agricultural statistics for the area in 2005 (Statistics Denmark, 2014), the agricultural area was split into 7 combinations of farm type and manure intensity, and each combination was represented by two 5-year crop rotation schedules, which were all incorporated in the SWAT model (Table 1).

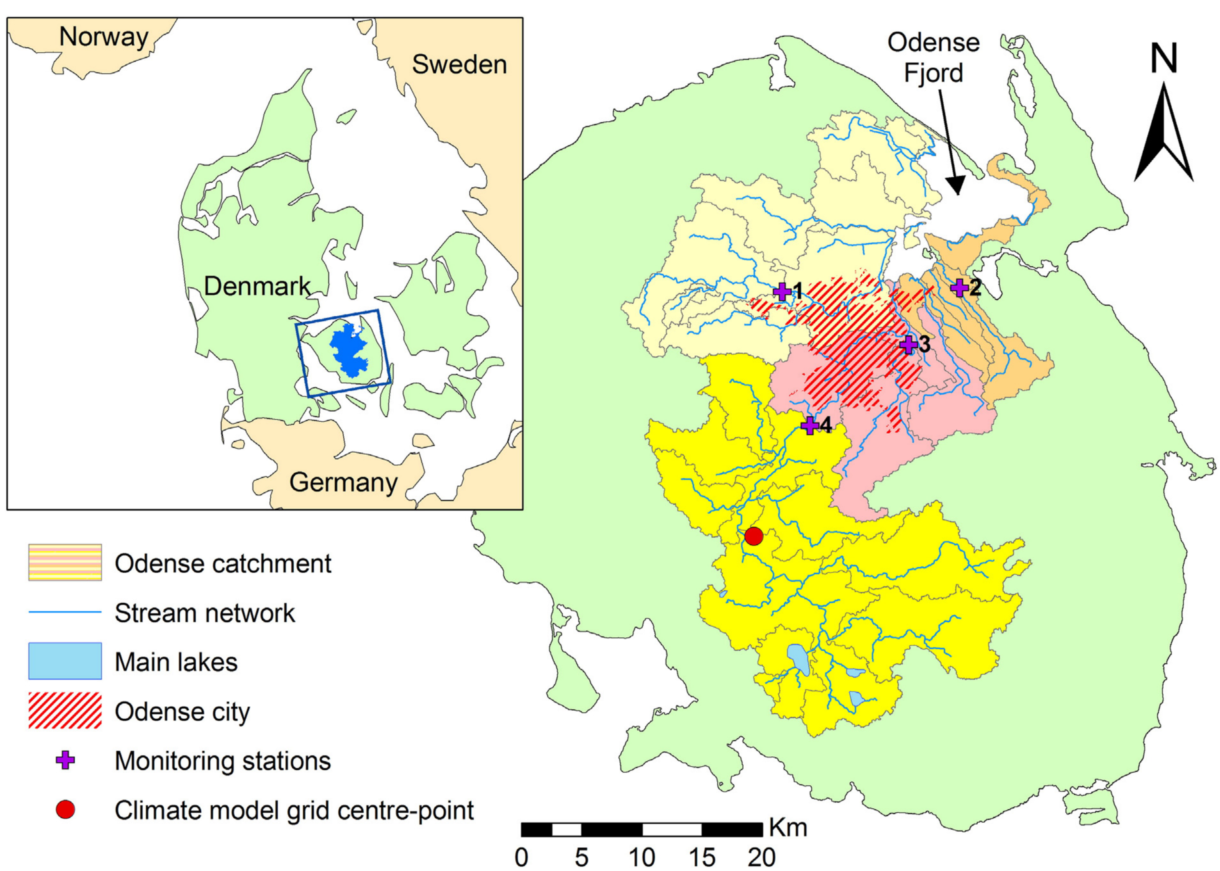


Fig. 1. Location of the Odense Fjord catchment, sub-basin division and stream network as represented in SWAT, as well as the location of runoff and nutrient monitoring stations and the centre point of the climate model grid used to derive local climate projections. Different sub-basin colours represent the clustering scheme for parameter transfer.

Table 1
Farm types and crop rotations used to describe agricultural management in the Odense catchment (W: winter, S: Spring).

Farm type	Manure N (kg N ha^{-1})	% Farm area	Rotation scheme				
			Year 1	Year 2	Year 3	Year 4	Year 5
Mixed and plant	<50	3.9	W. wheat	W. wheat	S. barley	W. wheat	W. wheat
			Seed grass	S. barley	Grass	W. wheat	Seed grass
Pigs	<70	32.0	W. wheat	W. wheat	W. wheat	S. barley	W. rape
			W. wheat - Grass	S. barley	Seed grass	Sugar beet	S. barley
	>70	28.0	W. rape	W. wheat	W. wheat	W. wheat	S. barley
			S. barley	Seed grass	S. barley	W. wheat	W. wheat
Dairy/cattle	<85	6.0	W. wheat	Seed grass	Sugar beet	S. barley	W. wheat
			S. barley	W. wheat	W. wheat - Grass	S. barley	Grass
	85–170	10.1	Maize silage	Maize silage	Maize silage	S. barley	S. barley - Grass
			W. wheat	W. wheat	Grass	Sugar beet	S. barley
	>170	3.9	S. barley	Maize silage	Maize silage	Maize silage	Maize silage
			S. barley	Maize silage	S. barley	Grass	S. barley
Mixed and horticulture	>50	16.1	Sugar beet	S. barley - Grass	S. barley	Seed grass	W. wheat
			Grass	S. barley	W. wheat	W. wheat -Grass	S. barley

Management operations were added, including individual fertilizer application per crop in each of the 14 rotations, distinguishing also between chemical fertilizers and manure (Thodsen et al., 2015). In total, 3410 Hydrologic Response Units (HRUs) – a unique combination of soil type, land use (including management) and slope class, which form the key building blocks and computational units in SWAT – were created.

Four monitoring stations were used for calibration and validation of both discharge and nutrient loads (Fig. 1). For all stations, daily data were available for flow, and for nutrients biweekly data were available for stations 1, 2 and 3, and every other day (at least) for station 4. Relative to the study by Thodsen et al. (2015), we updated the model to ArcSWAT 2012 rev. 582 and set up a new automatic calibration procedure with SWAT-CUP (v. 5.1.6) that included new parameters and accounted for phosphorus fractions. Calibration was performed on a daily basis from 1 Jan. 2000 to 31 Dec. 2005, with a 10-year warm-up period and split-sample validation (1 Jan. 2006–31. Dec. 2009, Refsgaard et al., 2014). The procedure ensured a very good graphical and statistical performance for flow calibration and validation. Detailed information about the calibration procedure can be found in Molina-Navarro et al. (2017). In this paper, the authors assessed the impact of the objective function when calibrating the four nutrient species also addressed in the present work (nitrate —NO_3^-—, organic nitrogen —OrgN—, phosphate —PO_4^{3-}— and organic phosphorus —OrgP—) simultaneously in this multi-site set-up. Following Molina-Navarro et al. (2017), the Nash-Sutcliffe efficiency coefficient (NSE, Nash and Sutcliffe, 1970) was chosen as objective function for the subsequent nutrient calibration because it yielded the best performance. A few improvements were done compared with Molina-Navarro et al. (2017): a new parameter, slope of the main channel, was included for the purpose of coherence (only slope of the tributaries was included before) and the initial range for the concentration of organic nutrients in the channel materials was expanded according to literature (Audet et al., 2011; Kronvang et al., 2012). Consequently, 14 and 20 parameters at basin-wide and sub-basin level, respectively, were calibrated for nutrients (Table 2). Once calibration was finished, ANION_EXCL, CH_BNK_KD and GWSOLP were manually tuned in some sub-basins to improve the calibration, while ensuring to keep the values inside pre-defined realistic ranges.

Model accuracy during nutrient calibration and validation was evaluated using a range of performance metrics including NSE, coefficient of determination (R^2) and percent bias (PBIAS). Visual inspection of simulated model output against the observed data was also undertaken since using only performance metrics can be misleading and produce statistically good but unrealistic simulations (Daggupati et al., 2015).

In order to perform the subsequent scenario simulations, transfer of the best parameter values to the non-calibrated sub-basins was needed. Accordingly, these were grouped in four clusters, each one corresponding to one of the calibrated sub-basins (Fig. 1). The clustering was done considering the position of each sub-basin in the river network and similarity of land cover, soil type and geology between the non-calibrated and calibrated sub-basins.

2.3. Scenario definition

Various future climatic and socio-economic scenarios were chosen within the context of the MARS project to define three storylines at European level, differing mainly relative to four main aspects: main drivers in the economy, economic growth, policies regarding the environment

Table 2
Initial range and calibrated values of the selected parameters (parameter acronyms with their meanings are listed in Appendix A. More detailed explanation can be found in Arnold et al., 2014).

Parameter	Level	Initial range	Calibrated values
CMN.bsn	Basin-wide	0.002–0.003	0.0027
PRF_BSN.bsn	Basin-wide	0.5–2	0.85
RSDCO.bsn	Basin-wide	0.02–0.1	0.058
SPCON.bsn	Basin-wide	0.001–0.01	0.0036
SPEXP.bsn	Basin-wide	1–1.5	1.18
ADJ_PKR.bsn	Basin-wide	0.5–2	1.09
CDN.bsn	Basin-wide	0.02–0.3	0.19
NPERCO.bsn	Basin-wide	0.21–0.95	0.58
N_UPDIS.bsn	Basin-wide	20–100	87.4
SDNCO.bsn	Basin-wide	0.8–0.88	0.85
PHOSKD.bsn	Basin-wide	140–450	180.8
PPERCO.bsn	Basin-wide	11–16	11.2
PSP.bsn	Basin-wide	0.01–0.7	0.19
P_UPDIS.bsn	Basin-wide	28–95	51.4
ANION_EXCL.sol	Sub-basin	0.1–1	0.40–0.87
BC1.swq	Sub-basin	0.1–0.99	0.27–0.85
BC2.swq	Sub-basin	0.2–2	0.90–1.52
CH_BED_KD.rte	Sub-basin	0.001–3.75	1.25–3.44
CH_BNK_KD.rte	Sub-basin	0.001–3.75	0.30–2.60
CH_D.rte	Sub-basin	0.58–3.57	0.81–3.02
CH_L1.sub	Sub-basin	10.83–32.21	11.92–30.89
CH_L2.rte	Sub-basin	6.10–21.51	7.30–18.56
CH_N1.sub	Sub-basin	0–0.4	0.05–0.30
CH_N2.rte	Sub-basin	0–0.4	0.01–0.29
CH_ONCO.rte	Sub-basin	500–20,000	3850–12,293
CH_OPCO.rte	Sub-basin	0–3000	435–2388
CH_S1.sub	Sub-basin	0.001–0.010	0.001–0.009
CH_S2.rte	Sub-basin	0.016–0.000	0.001–0.150
CH_SIDE.rte	Sub-basin	0–5	0.99–4.99
CH_W1.sub	Sub-basin	1.72–22.62	1.98–22.20
CH_W2.rte	Sub-basin	2.29–30.16	2.55–25.03
GWSOLP.gw	Sub-basin	0.01–0.2	0.05–0.08
HLIFE_NGW.gw	Sub-basin	0–600	431–549
USLE_P.mgt	Sub-basin	0–0.1	0.02–0.08

and public concern about the environment and protection of ecosystem services. A short description is provided here, but detailed information about the development of the storylines can be found in Faneca Sanchez et al. (2015).

- Storyline 1, called *Techno world* (*or economy rules*), considers that economy grows fast, with high energy demands and increase of CO_2 emissions due to a fossil-fuelled development. Society shows high awareness about environmental issues, but protection regulations are poor and policies are not renewed. Water management strategies are oriented to water needed for economic development and little effort is done regarding sustainable measures. Climate change scenario RCP 8.5, a rising scenario with very high greenhouse gas emissions, is assigned to this storyline.
- Storyline 2, called *Consensus world*, considers that the economy and the population are growing at the same pace as now. There are regulations to save energy in favour of reducing emissions, environmental awareness and interest for preservation. Current environmental guidelines and policies are continued but in a more integrated manner. First choice in water management strategies is sustainable at mid-long-term low-cost solutions. RCP 4.5 climate change scenario, a stabilisation scenario, was assigned to this storyline.
- Storyline 3, called *Fragmented world*, considers a regional rivalry in Europe where the economy grows in some countries (including Northern Europe) and decreases in others. Europe suffers from lack of resources and an extended use of fossil fuels. No attention is paid to the preservation of the ecosystems, although rich countries implement some mitigation measures. Environmental policies are violated because of focus on economic development. Water management actions are restricted to short-term effects. Again, climate change scenario RCP 8.5 rules in this storyline.

We downscaled the above three storylines to the Odense Fjord catchment in order to simulate future scenarios at catchment scale. The time horizons were set to 2030 (interval 2025–2034) and 2060 (2055–2064) – 2030 due to the update to 2027 as the deadline for fulfilling WFD targets (one of the objectives of MARS is to support managers and policy makers in practical implementation of the update of the WFD) and 2060 to show the impacts of climate change as climate projections show little change in climate variables from todays until 2030 (Faneca Sanchez et al., 2015)–.

Regarding future climate data, the MARS project chose to use the ISI-MIP project climate scenarios (www.isi-mip.org). The ISI-MIP project has produced data for climate scenarios using five different climate models (GFDL-ESM2M, HadGEM2-ES, IPSL-CM5A-LR, MIROC-ESM-CHEM and NorESM1-M). Within MARS, the five models were compared regarding cumulative precipitation (2006–2009) at 0.5° spatial disaggregation for both RCPs, 4.5 and 8.5. An extra check was done by determining the cumulative river runoff per model and selecting the median model based on the outcomes of the global model PCR-GLOBWB (PCRaster Global Water Balance, www.globalhydrology.nl/models/pcr-globwb-2-0). In the Odense Fjord catchment, the IPSL-CM5A-LR model yielded the best median output, regarding both cumulative precipitation and cumulative runoff, relative to observations (MARS internal document); thus, this model was selected to simulate climate change in the catchment. IPSL-CM5A-LR data were also used to run two baseline scenarios (one per RCP) covering the period 2011–2020. Projected (and not observed) climate data were used as baseline in order to obtain fully comparable results with the future storyline scenarios. Moreover, the period 2011–2020 was chosen to be consistent with the 10-year interval chosen for the scenario simulations and to centre the baseline on the year 2016. In this way, the future scenarios will show changes relative to today, which is optimum for water management purposes.

The variables selected to force the climate change scenarios in SWAT were maximum and minimum air temperature, precipitation, solar radiation, relative humidity and wind speed. Climate change data were obtained for the period 2006–2099 from the ISI-MIP project, which offers daily projections for the required variables in grid cells of 0.5°. The network point located inside the Odense catchment (lat 55.25, long 10.25, Fig. 1) was selected to extract the daily projections. Table 3 shows the annual average changes projected by the IPSL-CM5A-LR climate model (RCPs 4.5 and 8.5) for the included climate variables. Changes in detailed monthly projections are given in Appendix B.

The projections show increases in precipitation on an annual scale, slight in the RCP 4.5 and more noticeable in the RCP 8.5 scenarios (up to 13% in the long-term scenario, Table 3). Regarding seasonality, no clear pattern emerges in the RCP 4.5, while in RCP 8.5 precipitation increases in winter and autumn and decreases in summer, especially in the long-term horizon (see Appendix B). For temperature, the projections demonstrate an increase, lower during the first term (between 0.7 and 1.1 °C) and higher during the second term (between 1.4 and 2.8 °C, Table 3). The temperature increases are higher in winter (see Appendix B) as demonstrated by an up to 4.7 and 5.3 °C increase in maximum and minimum temperatures in February in RCP 8.5 in the long-term horizon. Relative humidity decreases across scenarios, following a trend opposite to that of temperature (Table 3). Average annual changes in solar radiation and wind do not show a particular trend. However, the projected monthly changes often show an opposite behaviour between projected solar radiation and precipitation (see Appendix B). Additional information about the IPSL-CM5A-LR climate model can be found at IPSL Climate Modelling Centre (2017).

Before running the scenarios, we compared the first years of the projections (2006–2014) with observed data and applied the appropriate bias correction for precipitation and temperature projections on a monthly basis through the ArcSWAT interface (Table 4). The model interface requires a mean temperature correction, so the average of the monthly bias for both maximum and minimum temperatures was calculated.

Regarding land use change, since our study area is predominantly occupied by agriculture (68% of the catchment), we focused on farming to design three scenarios corresponding to the storylines:

- Storyline 1. High-tech agriculture (HT): The size of the agricultural area remains unchanged, with some conversion to permanent grass and willow. There is a slight increase in livestock density and a slight decrease in application rates of artificial fertilizer.
- Storyline 2. Agriculture for nature (AN): The agricultural area decreases and changes towards forest and less intensive farming types. Artificial fertilizer application decreases slightly.
- Storyline 3. Market-driven agriculture (MD): The agricultural area increases and changes towards intensive pig and dairy farms. Livestock density and fertilizer application increase.

The scenarios were based on a comprehensive study done by the Centre for Regional change in the Earth System (CRES, Olesen et al., 2014), and the details of the applied changes mentioned above can be

Table 3
Projected changes (annual averages) relative to the baseline period (2011 − 2020) for the climate variables forced in SWAT according to the IPSL-CM5A-LR climate model.

Period	2025–2034		2055–2064	
Scenario	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Precipitation (%)	0.6	5.6	1.9	13.2
Max. temp. (°C)	0.7	0.9	1.4	2.5
Min. temp. (°C)	0.9	1.1	1.7	2.8
Rel. humidity (%)	−0.1	−0.3	−0.3	−0.8
Solar radiation (%)	−0.3	0.7	0.2	−1.7
Wind speed (%)	3.9	−0.5	2.0	3.0

Table 4
Monthly bias correction applied to the IPSL-CM5A-LR climate model (RCPs 4.5 and 8.5) after comparing projected and observed data from the period 2006–2014.

	Temperature bias (°C)		Precipitation bias (%)	
	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Jan	−1.2	−0.7	26.5	40.0
Feb	−1.7	−1.1	19.4	35.3
Mar	−0.5	−0.3	−25.5	0.6
Apr	1.1	1.4	−17.5	−16.3
May	0.8	1.0	30.7	17.5
Jun	−0.5	0.1	63.2	53.0
Jul	0.7	0.7	23.9	6.0
Aug	0.1	0.1	85.1	87.3
Sep	0.5	0.0	−0.1	13.6
Oct	−0.3	0.3	18.3	41.0
Nov	−0.3	0.3	−1.0	0.7
Dec	−0.3	−0.4	50.8	67.1

Table 6
Calibration (2000–2005) and validation (2006–2009, in brackets) performance statistics values for daily nutrient loads at the four monitoring points in the Odense Fjord catchment (NSE: Nash-Sutcliffe efficiency, R2: Coefficient of determination, PBIAS: Percent bias; positive and negative PBIAS values indicate model under- and overestimation, respectively).

	Station 1	Station 2	Station 3	Station 4
		NO_3^-		
NSE	0.71 (0.40)	0.72 (−2.59)	0.69 (0.42)	0.64 (0.64)
R2	0.71 (0.66)	0.80 (0.09)	0.70 (0.49)	0.64 (0.75)
PBIAS	−1.5 (−45.6)	−3.0 (−62.9)	1.1 (−6.6)	−1.4 (−26.9)
		OrgN		
NSE	0.25 (−0.78)	−1.34 (−1.41)	0.32 (0.45)	0.39 (0.16)
R2	0.53 (0.62)	0.00 (0.00)	0.45 (0.50)	0.46 (0.41)
PBIAS	1.4 (−31.5)	98.2 (98.5)	−7.8 (−17.7)	6.6 (3.3)
		PO_4^{3-}		
NSE	0.34 (0.24)	0.51 (0.44)	−0.20 (−0.21)	0.22 (0.11)
R2	0.37 (0.42)	0.51 (0.46)	0.00 (0.04)	0.27 (0.23)
PBIAS	6.0 (−34.6)	4.3 (−7.0)	3.9 (−8.7)	6.0 (−1.4)
		OrgP		
NSE	0.42 (−0.41)	−1.05 (−1.59)	0.46 (−0.02)	0.33 (0.17)
R2	0.54 (0.33)	0.01 (0.02)	0.46 (0.48)	0.41 (0.43)
PBIAS	7.4 (−20.7)	89.6 (90.5)	2.0 (−29.3)	−0.9 (−7.7)

found in Appendix C. For the long-term scenario, land use changes were applied in SWAT through changes in land use type area, in fertilizer and manure application rates and in timing of crop management operations (Tables C1 and C2 and Fig. C1). In the short-term scenario, timing was kept as in the baseline, while the dates for willow, added in the HT scenario, were kept similar in both periods (manure and fertilizer application on 1st April, harvest on 1st December). A warmer climate would allow grain maize production in Denmark; therefore, this crop was introduced in all long-term land use change scenarios, replacing a portion of the winter wheat (Table C3 compared with Table 1), with the same fertilization rates. All these changes led to a global decrease in the total amounts of nitrogen and phosphorus applied in fertilizer in HT and AN and an increase in MD (Fig. C2).

In all, 12 scenarios were run in SWAT (Table 5):

- 4 scenarios with the last ten years of observed climate data (2001 — 2010), one with the current land use and three with the future land use scenarios. This procedure is necessary to evaluate the isolated effects of land use changes on hydrology and nutrient load, since in the storylines each land use is run together with different climate models/RCPs.
- 2 scenarios that act as baseline for the MARS storylines, one per RCP centred in the present time (2011–2020) and with the current land use.
- 6 scenarios to simulate the described three future storylines in two time horizons: 2030 (interval 2025–2034) and 2060 (interval 2055–2064).

3. Results

3.1. Model calibration and validation

The initial and calibrated values of the selected parameters for nutrient calibration are given in Table 2. The observed and simulated nutrient fraction exports at the monitoring stations during calibration and validation periods are provided in Appendix D, and Table 6 shows the corresponding performance statistics values.

3.2. Scenario simulations

3.2.1. Effects of isolated land use change scenarios

To investigate the effects of isolated land use change scenarios (LUC) on discharge and nutrient loads, LUC scenarios were run with the same (observed, 2001 to 2010) climate. Regarding water balance components, actual evapotranspiration (AET) was very similar in all the scenarios, exhibiting a slight decrease in HT. Water yield (Q) slightly increased in HT and AN but remained unchanged in MD. The change in soil water content (SWCC, the difference between SWC at the beginning and at the end of the simulation) did not vary among the scenarios (Fig. 2). The changes in other components of the water balance (snow sublimation, transmission losses and variation of the aquifer storage) were minor and did not show variations between the scenarios (data not shown).

Table 5
Abbreviations used for the 12 scenarios run in SWAT to analyse the individual effect of land use (a) and the possible effects of the MARS storylines (b).

a)

	Isolated land use change scenarios			
Climate	Present land use	Agriculture for nature	High tech agriculture	Market-driven agriculture
Observed	PLU_Obs	AN_Obs	HT_Obs	MD_Obs

b)

	MARS storylines: Land use + climate change scenarios		
Time horizon	Agriculture for nature (Consensus world) + RCP 4.5	High tech agriculture (Techno world) + RCP 8.5	Market-driven agriculture (Fragmented world) + RCP 8.5
Baseline	PLU_45	PLU_85	
2030	AN_30	HT_30	MD_30
2060	AN_60	HT_60	MD_60

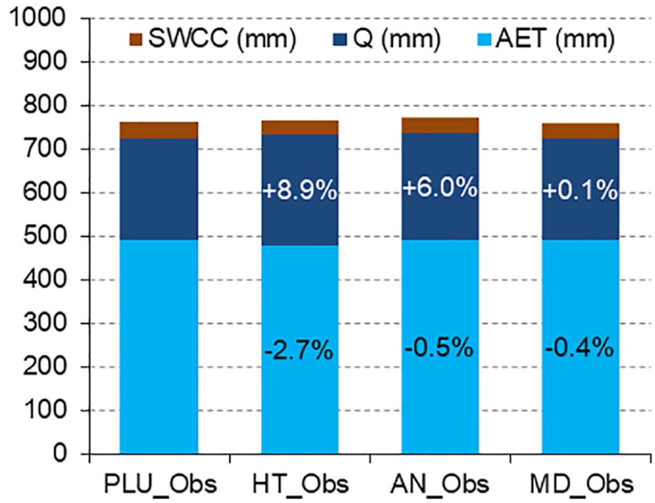


Fig. 2. Actual evapotranspiration (AET), water yield (Q) (average annual values) and change in soil water content (SWCC) in isolated land use change scenarios under observed climate trends. Column labels show the percent change from the baseline (PLU_Obs) for AET and Q (not shown for SWCC since absolute changes were negligible).

Changes in flow components differed among the LUC scenarios (Fig. 3). The contribution by groundwater to stream flow was always dominant, but it increased in HT and AN, decreasing tile drain flow in these scenarios, while the opposite was observed in MD. Surface and lateral flow contributions remained similar in all the scenarios.

Organic N and P loads were slightly higher in HT than in the other scenarios. The NO_3^- load decreased noticeably in HT and even more in AN, while it increased in MD. The PO_4^{3-} load showed an increase in HT and AN, the increase being slightly higher in HT than in AN (Fig. 4).

3.2.2. Simulations of MARS storylines

Since we simulated scenarios departing from different baselines (projected climate with different RCPs, 8.5 for HT and MD, 4.5 for AN), the results are presented as absolute changes in each scenario relative to its baseline (Table 5b) and not as total values. This we do in order not to confuse the reader and to facilitate the subsequent discussion.

Regarding the water balance, AET increased in all the scenarios. SWCC decreased in all the scenarios relative to the baseline. Flow (Q) increased in HT and MD but remained unchanged in AN (see Fig. 5, showing also the changes in precipitation). Variations in minor components of the water balance (e.g. aquifer recharge) were not relevant.

Table 7 shows the relative contributions of flow components across the different scenarios. A pattern similar to that in the LUC scenarios was observed for both time horizons, with MD having lower groundwater and higher tile drain flow contributions than HT and AN. Fig. 6 shows the absolute changes from the baseline in tile drain flow and groundwater flow. The absolute changes in surface and lateral flow between scenarios were minor (maximum absolute variations of annual averages from the baseline were 1.8 mm and 1.6 mm, respectively).

Fig. 7 depicts the expected changes in nutrient loads. While organic nutrients exhibited a similar trend, variations in predicted loads differed for the two mineral species modelled.

4. Discussion

4.1. Model calibration and validation of nutrient exports

The dynamics of the different nutrient fractions were adequately represented by the model in most cases (Figs. D.1–D.4). The statistical performance also reflects successful calibration (Table 6) relative to the criteria by Moriasi et al. (2007), especially when considering the facts that the model was calibrated with daily values, usually resulting in lower performance than when using monthly time steps (Gassman et al., 2007), and that it was a spatial (4 stations) and multi-variable (4 nutrient species) calibration, rendering it much more difficult to obtain satisfactory results for all the variables (White and Chaubey, 2005). Unsatisfactory results were, though, obtained for organic N and P at station 2 (Fig. D.2), which belongs to the smallest among the monitored sub-basins ($\approx$26 km^2) and is located on the border of the catchment (Fig. 1). An explanation may be that the discharge is too low for SWAT to produce a realistic estimation of the organic nutrient load (which in a lowland catchment may derive mainly from riverbank erosion or stochastic riverbank collapses). Also, NSE and R^2 values were unsatisfactory for PO_4^{3-} at station 3, the volumes being correct, as demonstrated by the good PBIAS, but the timing of peaks wrong. NO_3^- calibration for stations 2 and 3 also had some outlying values that were much higher than the corresponding observed values.

During the validation period, the model mainly showed a good reproduction of the observed nutrient loads, keeping the above-mentioned imprecisions (Figs. D.1–D.4). Statistical performance metrics decreased during validation (lower R^2 and NSE, higher absolute PBIAS), which is expected for a highly parameterised model such as SWAT (Gassman et al., 2007). The much poorer statistical performance for NO_3^- at station 2 (Table 6), despite a satisfactory graphical performance similar to the calibration period (Fig. D2), is due to the simulated outlying values; the highest outlying value (10th July 2006) matched with an observation point, and if this was excluded the calculated performance metrics would increase substantially (NSE $= 0.62$, $R^2 = 0.80$, PBIAS $= -41.6$). Nevertheless, the majority of the values are acceptable for a multi-site and multi-variable daily calibration. With the exception of the organic fractions as in station 2, absolute PBIAS is always lower than 70, in most cases lower than 40 and frequently lower than 25, which indicates “satisfactory”, “good” or “very good” performance of a monthly calibration (Moriasi et al., 2007).

4.2. Scenario simulations

4.2.1. Effects of land use change scenarios

Changes in land use alone had only a minor effect on the water balance components (Fig. 2), which was probably related to the variation of the tile drained surface or the simulated location of the new land

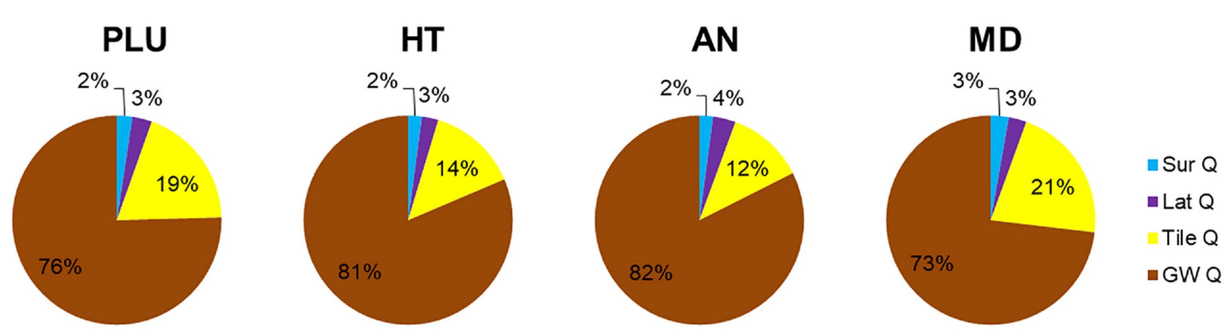


Fig. 3. Percent contribution of flow components in isolated land use change scenarios with observed climate data (Sur: surface, Lat: lateral, Tile: Tile Drain, GW: Groundwater, Q: flow).

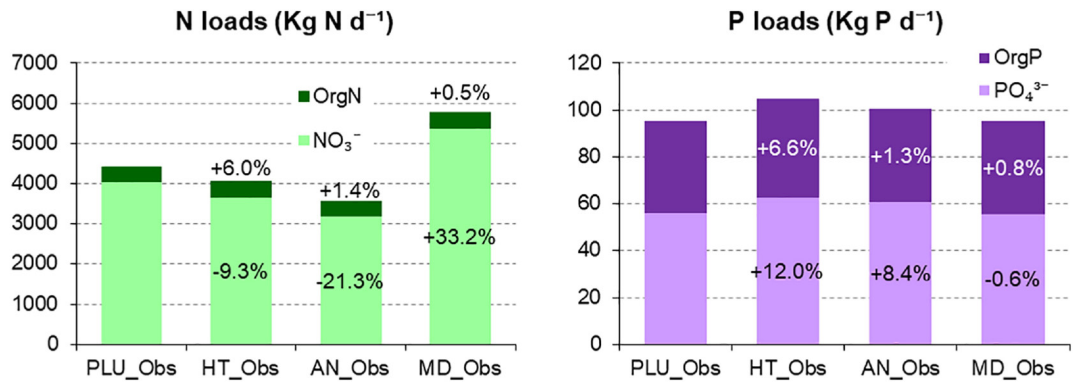


Fig. 4. Nutrient loads in isolated land use change scenarios with observed climate data. Column labels show the percent change from the baseline (PLU_Obs) for each nutrient fraction.

uses in the catchment (e.g. different soil properties). Regarding flow components, tile drain flow decreased in the HT and AN scenarios (Fig. 3) due to the conversion of tile drained agricultural areas to non-tile drained forested areas (Table C1), increasing groundwater flow. The opposite trend, but less pronounced, was observed in the MD scenario (Fig. 3), where there was an increase of the currently farmed surface (Table C1).

Regarding nutrients, the NO_3^- load changes followed exactly the same pattern as the variation in fertilization in the scenario simulations (Figs. C2 and 4). NO_3^- added via fertilization is quickly taken up by plants, increasing the plant (and root) biomass and therefore also the organic matter pool that is eventually mineralised into NO_3^- and leached. Given the similar patterns for NO_3^- load and fertilization, the conclusion is that fertilization is the main responsible factor determining the NO_3^- load in the catchment. However, changes in the loads of OrgN, OrgP and PO_4^{3-} did not follow the same pattern but showed slight variations. Similar observation have been made in other SWAT analyses revealing fertilization to be an effective measure to reduce N exports but not P exports (Molina-Navarro et al., 2014; Panagopoulos et al., 2011). This is because P is not lost directly from fertilizer application but from the soil P pool via soil erosion and surface runoff; also, a large proportion is lost through bank erosion (Kronvang et al., 2012). PO_4^{3-} load showed a variability similar to that of the groundwater flow, with a small increase in HT and AN and a slight reduction in MD (Figs. 3 and 4), which may indicate that groundwater is the main source of PO_4^{3-} in the Odense catchment. Changes in organic nutrient loads were minimal, showing only a slight increase in HT (Fig. 4), which did not correspond with the variations in fertilization. In SWAT, organic nutrients are associated with sediment transport (Neitsch et al., 2011) and higher sediment transport into the estuary was actually found in HT (389 T d^{-1}) than in the other scenarios (371, 376 and 378 T d^{-1} in PLU, AN and MD, respectively). This might be related to the higher total flow since an increase of annual river discharges could increase channel erosion (Thodsen, 2007).

To sum up, the isolated land use change scenarios had an effect on NO_3^- loads via the changes in fertilization, while the changes in loads of the other three nutrient fractions were less pronounced and appeared related to changes in flow and in-stream sediment change.

4.2.2. Simulations of MARS storylines

AET increased in all the scenarios (Fig. 5), following the pattern of temperature change (Table 3), but the change was small compared with those of the other balance components, being 1–3% higher in the short term and 4–6% higher in the long term. The AET increase was the main driver of the small Q decrease in the AN scenarios where precipitation increased only slightly (Fig. 5). In the HT and MD scenarios, however, precipitation was projected to increase much more, especially in the long term, resulting in an overall increase in discharge despite the increase in AET (Fig. 5). The main reason of these differences is the use of two different concentration pathways, 4.5 for AN and 8.5 for HT and MD, especially due to their different precipitation projections (Fig. 5). Higher flow in HT than in MD despite the use of the same climate projections might be due to the land use effect, as shown in Fig. 2.

The soil water content increased in all scenarios; the absolute change (SWCC) being lower (decreased) in the simulated scenarios than in the baseline, though (Fig. 5). This was because, despite that the soil water content at the end of the simulation was very similar in all the scenarios (between 215 and 219 mm, data not shown), the soil content at the beginning of the simulation was higher in the future scenarios (18–32 mm higher) than in the baseline. The projected increase in the precipitation in HT and MD might be the main driver of the higher initial soil water content in these scenarios (Sellami et al., 2016), yielding a lower soil water content change. Canopy cover may also influence the soil water regime by offsetting drier conditions (Holsten et al., 2009). The forest surface increased in AN, which might explain the higher initial soil water content – decreasing SWCC – in this scenario despite the only small projected increase in precipitation (Fig. 5). Tian et al. (2016) also found an increase in soil water storage when simulating conversion

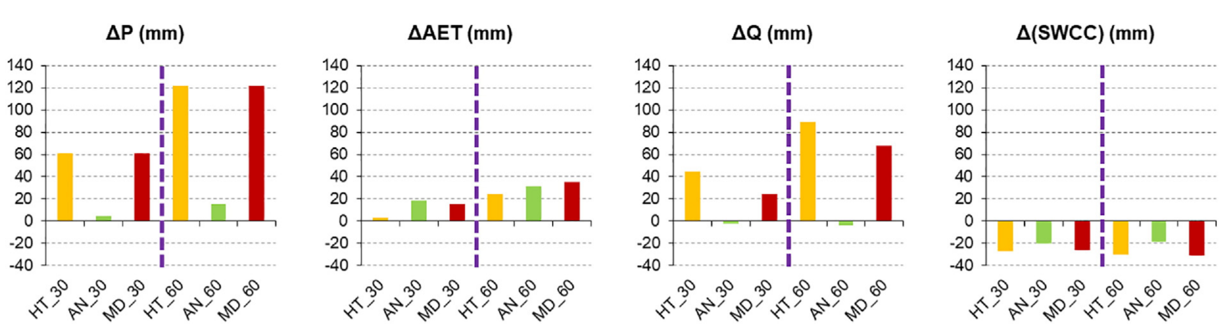


Fig. 5. Predicted absolute variations (scenarios vs. baseline) of four water balance components: precipitation (P), actual evapotranspiration (AET), flow (Q) (annual average values difference) and soil water content change (SWCC, difference in the absolute water change at the end of the simulation).

Table 7
Contribution (%) of the different flow components to total discharge across scenarios (Sur: surface, Lat: lateral, Tile: Tile Drain, GW: Groundwater, Q: flow).

	PLU_45	PLU_85	HT_30	AN_30	MD_30	HT_60	AN_60	MD_60
Sur Q	3.2	3.2	1.9	2.0	2.3	1.8	2.3	2.1
Lat Q	3.7	4.0	3.2	4.5	3.5	2.9	4.6	3.2
Tile Q	20.1	20.2	15.0	12.7	23.4	14.2	12.6	21.9
GW Q	73.0	72.6	79.9	80.7	70.8	81.0	80.4	72.8

of grass or shrub to forest using SWAT. Yet, other authors have found opposite effects of forest coverage on soil water change in a changing climate (Holsten et al., 2009); thus, the results should be interpreted with caution.

In all scenarios, LUCs had only a minor effect on the hydrological response, which has previously been observed by other authors (e.g. Karlsson et al., 2016; van Roosmalen et al., 2009). However, the possibility that LUCs may modulate catchment hydrology (both total discharge and its components), with a subsequent effect on nutrient loads, should not be disregarded. The change of crop management dates in the 2060 time horizon might also affect the hydrological dynamics, although to a minor degree than other drivers. In another Danish catchment van Roosmalen et al. (2009) found that changes in cropping dates only had a slight effect on the mean annual recharge, which increased 5 mm as the result of an earlier increase in the soil moisture content due to earlier harvesting of grains. In our study, harvesting dates also occurred earlier in the 2060 time horizon (Fig. C1), implying that a similar effect may be expected. It should also be noted that due to the temperature increase projected in all scenarios (Table 3), SWAT projected a noticeable decrease in snowfall, up to 45% for 2030 and as much as 81% for 2060, which will also affect the river dynamics.

Groundwater remained the dominant flow component. The changes in the relative contributions of groundwater and tile drain flow observed for LUC alone were similar for the three future storylines with corresponding LUC (Table 7); however, absolute variations differed between the scenarios due to different climate change inputs. In AN, since total flow remained unchanged, flow components varied as observed when simulating LUC alone, with decreasing tile drain flow and increasing groundwater flow (Fig. 6). In HT, however, total flow increased, and absolute tile drain flow remained similar despite the reduction of tile drained agricultural areas, meaning that flow per tile drained surface unit increased. This corresponds with the findings of van Roosmalen et al. (2009), who anticipated enhanced drain flow in climate change-only scenarios due to higher groundwater levels. Thus, in absolute terms, the flow increase in HT could be ascribed entirely to the groundwater flow (Fig. 6). In MD total flow increased due to an increase of both tile drain and groundwater flow (Fig. 6), while in the LUC-only scenarios tile drain flow increased and groundwater flow decreased.

Our findings supplement those of previous model investigations undertaken in similar areas. Most of these have predicted a runoff increase (e.g. Thodsen, 2007; Thodsen et al., 2014; van Roosmalen et al., 2009), some of them using an extreme climate scenario (Trolle et al., 2015) that yielded results similar to our RCP 8.5 findings. However, our results also showed that runoff remained stable in a lower emission scenario, the difference between AN and the other storylines being a result of using a different RCP (4.5), as the isolated LUC was seen to slightly increase flow in AN. Earlier investigations have pointed to a precipitation increase, especially during winter, as the main reason for a future higher discharge, but in our study considering two emission pathways and time horizons a clear trend towards this was not obvious (Appendix B). Contrary to the findings in our study, Karlsson et al. (2015) found an overall increase in discharge for the RCP 4.5 scenario in the upper Odense catchment. However, they also simulated an idealised high-end 6 °C scenario and, despite increasing precipitation (+7%), they found a decrease in discharge, mostly due to a strong increase of AET (+17%) that dries out the soil. This result is opposite to the findings in our and the above-mentioned studies that predicted runoff increases at high emissions. It should be noted, though, that Karlsson et al. (2015) used a different climate model (RCM HIRHAM5) and another hydrological model (MIKE SHE), which again emphasises the relevance of model choice in future simulations or considerations regarding multi-model ensemble approaches. They also used a different target (2071–2099) and historical control period (1991–2005). The fact that our work uses a control – baseline – period centred in the present (2011–2020) is a novelty since the control periods in previous research studies usually are the 90s and early 00s, or even earlier.

The analysis of nutrient loads results becomes complex when climate change effects and land use changes are combined. Regarding organic nutrients, N and P followed the same pattern (Fig. 7), resembling closely that of the variation in total flow (Figs. 5 and 6) and the changes in sediment transport (data not shown), as observed when analysing solely LUC. Thodsen et al. (2008) have predicted sediment transport to increase in the Odense River in a warmer and wetter future climate, especially in winter, as a consequence of increased precipitation and river discharge (as in the HT and MD scenarios), and other studies have suggested that total nutrient losses in the future are related to runoff variations (e.g. Øygarden et al., 2014, regarding N); however, to the best of our knowledge, ours is the first study to address organic nutrients specifically in northern Europe. Also the PO_4^{3-} variation within the scenarios exhibited a pattern very similar to those of the groundwater flow (Figs. 6 and 7), which confirmed that groundwater flow could be the main PO_4^{3-} source in the Odense catchment.

Since the effects of LUCs on hydrology were minor, climate changes – conditioned by the use of two scenario-specific RCPs –, were the main factor driving the future changes in organic nutrients and PO_4^{3-} loads. The loads of both OrgP and PO_4^{3-} increased considerably in the HT and MD storylines (RCP 8.5) (Fig. 7). The increase was already pronounced in the 2030 scenarios (up to 34% for OrgP in HT) and even higher in the 2060 scenarios (up to 71% for OrgP in HT). In the AN storyline (RCP 4.5), OrgP slightly decreased and PO_4^{3-} slightly increased (Fig. 7), leading to a stable future concentration of TP. This result contradicts previous findings that mostly predicted an increase in TP (Trolle et al., 2015), as happened to stream flow. High phosphorus loads will threaten the health of the aquatic ecosystems of the catchment, especially

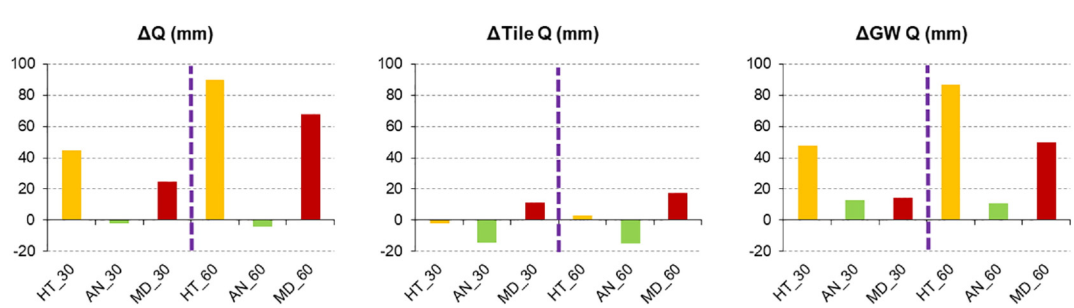


Fig. 6. Predicted absolute changes from the baseline in tile drain flow and groundwater flow (average annual values) for the different scenarios. The change in total flow (average annual values) is shown for the purpose of comparison.

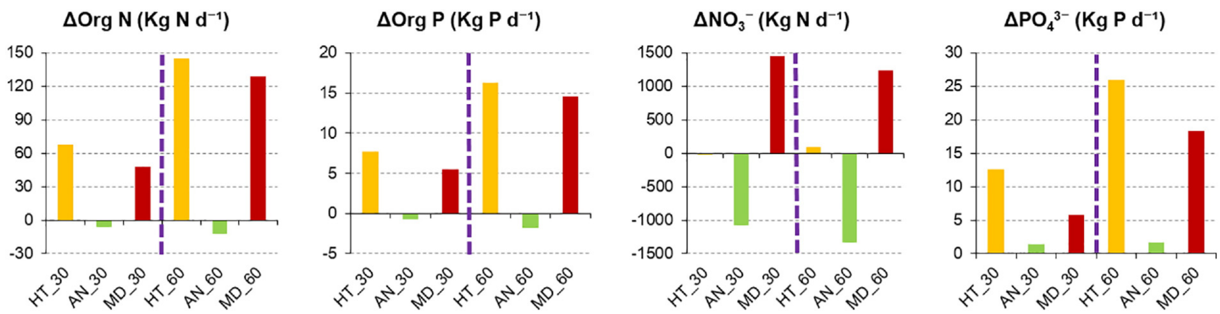


Fig. 7. Predicted absolute changes from the baseline in nutrient loads for the different scenarios.

Odense Fjord, which is already characterised by a moderate/bad ecological status (Miljø- og Fødevareministeriet, 2016b). This calls for the introduction of measures to reduce the phosphorus loads in river discharge, potentially with focus on controlling bank erosion, which is an important source of phosphorus in lowland catchments (Lu et al., 2015).

Similar to the analysis of the effects of LUC alone, fertilization (Fig. C2) had a strong influence on the NO_3^- load. However, the final NO_3^- export was a combination of effects of both stressors, LUC and climate change, and different climate inputs ultimately modulated the NO_3^- loads (Fig. 7). The HT and AN storylines both supposed a reduction of the fertilizer input (Table C2, Fig. C2) but a different climate input (different RCP). In the AN scenarios, where flow remained constant, the NO_3^- load decreased, whereas it was similar to the baseline in the HT scenarios because the decrease in fertilization was counteracted by flow. When differentiating between the contribution of NO_3^- from the different flow components, the NO_3^- load via tile drains decreased due to lower fertilization, whereas the NO_3^- load in groundwater flow increased since higher flow allows higher NO_3^- transport in spite of the lower fertilization. In contrast, the NO_3^- export increased in the MD scenarios where both flow and fertilization were higher.

The expected temperature rise could also yield higher NO_3^- losses due to increased mineralisation (Øygarden et al., 2014), although this may be counteracted by a further decrease of the soil organic matter pool in a constant warming up scenario, which will ultimately result in a decrease in NO_3^- leaching rather than an increase due to reduced mineralisation (Jensen and Veihe, 2009). Thus, although a future climate might favour higher N leaching and exports (Jensen and Veihe, 2009; Øygarden et al., 2014; Trolle et al., 2015), our results showed that this ultimately depends on the future LUCs in the study area. Actually, our results revealed that changes in management practices (i.e. change in fertilization) and not a shift in crops were the most important factor for changing the NO_3^- export, as hypothesized by Trolle et al. (2015). Moreover, a warmer climate creating changes such as a longer growing season, different operation dates and new crops may have additional effects on N losses (Øygarden et al., 2014). These changes were considered in our study, meaning that their effects were taken into account in our simulated final N loads, unlike in most other nutrient simulation studies. However, such changes are very difficult to isolate when simultaneously simulating all potential stressors in complex scenarios. Changes in precipitation may also influence the atmospheric deposition of N (Hessen, 2013). Actually, our models predicted a 7% and 14% (2030 and 2060, respectively) increase in NO_3^- deposition with rainfall in the HT and MD storylines as precipitation is projected to increase, while we assumed a constant NO_3^- concentration in the rainfall.

The organic contribution of N is much lower than that of NO_3^-, whereas the organic and mineral contributions of P are similar (Fig. 4). Thus, when considering mitigation of N, attention should be focused on the mineral fraction and its ultimate effects on the aquatic ecosystems. Our results on the impacts of the future climate and agricultural practices on N losses are of great interest since Denmark is one of the world's most intensively farmed countries (Kaspersen et al., 2016); in fact, agricultural production is estimated to contribute around 70% of the N entering the Danish coastal waters. A reduction of nutrient losses is therefore crucial for successful implementation of the WFD (Bouraoui and Grizzetti, 2014), and although Denmark is one of the most successful EU countries at reducing the nutrient loading of the aquatic environment, even further reductions are required (Kaspersen et al., 2016). Thus, simulation results like those presented in this paper should ideally be taken into account in the elaboration of the new river management plans to identify the most appropriate measures (e.g. Kaspersen et al., 2016; Øygarden et al., 2014) to limit the nutrient loads entering aquatic ecosystems.

4.2.3. Uncertainties

Even though highly detailed inputs were used in this study to ensure the reliability of results, some uncertainties remain and the results should therefore be interpreted with caution. The main source of uncertainty arises from the projected climate data, especially precipitation, as already acknowledged by many authors using projections from a single climate model (e.g. Jensen and Veihe, 2009; Trolle et al., 2015; van Roosmalen et al., 2009). To favour reliable predictions in this study, daily projections (not idealised values) of five meteorological variables and two RCPs were used, and two scenario horizons (2030 and 2060) with a control period centred in the present (2011–2020) were simulated. Moreover, the direct coupling between the hydrological model and the climate models promotes accuracy in the simulation of the combined effects of climate and land use changes (van Roosmalen et al., 2009). The hydrological model and the extrapolation of the climate projections from a 0.5° grid may result in an additional error (Karlsson et al., 2016; Thodsen, 2007), but this is minor relative to the uncertainty of the climate model.

Several authors (e.g. Arheimer and Lindström, 2015; Karlsson et al., 2016; Thodsen et al., 2014) have pointed out the importance of the choice of climate model in the assessment of future climate and hydrological impacts, and use of an ensemble approach is thus recommended. In this study, we chose the IPSL-CM5A-LR model because it yielded de best median output – both in terms of cumulative precipitation and runoff – out of five climate models compared by the MARS team (MARS internal document), suggesting that it might give the best representation of the future behaviour of climate in Denmark.

Some uncertainty might also arise from the lack of correction to account for the effects of increased CO_2 in the stomatal conductance, which might ultimately reduce evapotranspiration (Karlsson et al., 2015; van Roosmalen et al., 2009). However, the validity of this correction has been questioned, especially for high-end scenarios (Karlsson et al., 2016), since the extent of the CO_2 effect and possible feedback is highly uncertain and still debated in the scientific community (e.g. Peng et al., 2014; Zhu et al., 2012).

5. Conclusions

The SWAT model was successfully calibrated for stream discharge and loads of four nutrient fractions (OrgN, OrgP, PO_4^{3-} and NO_3^-), and

three future storylines were downscaled to the Danish Odense Fjord catchment with focus on changes in climate, land use and agricultural management. The future scenarios involved adaptation to a warmer climate through changes in crops, fertilization levels and timing of field operations.

Changes in land use alone affected total catchment runoff, albeit only to a small degree. The contribution of the groundwater flow to stream flows increased at the cost of the contribution of tile drain flow in the high technology (HT) and agriculture for nature (AN) scenarios, and vice versa in the market-driven agriculture (MD) scenario. Land use change did not markedly affect the losses of organic nutrients and of PO_4^{3-}; however, the increased N fertilization in the MD scenario resulted in a considerable increase in the loss of NO_3^- to surface waters. The opposite result was found in the HT and AN scenarios with decreased fertilization.

When introducing climate change on top of land use changes, it becomes apparent that the different RCPs used in the scenarios are crucial for the outcome: runoff increases in HT and MD (RCP 8.5), which corresponds with previous findings, but remains stable in AN (RCP 4.5).

The analyses of combined land use change and climate change showed that river discharge (and its individual flow components) was the main driver of changes in the loads of organic nutrients and inorganic P, the discharge variations mainly being influenced by the different climate inputs used. Both organic P and PO_4^{3-} loads increased considerably in the high emission scenario, which would threaten the ecological status of the aquatic ecosystems in the catchment, including Odense Fjord.

NO_3^- is the main N fraction in the catchment and both stressors, land use change (via inputs of fertilizer) and climate change, impacted the transport of NO_3^-. Different levels of fertilization had a strong influence on the inorganic N load, which decreased in AN and increased in MD. However, in HT, NO_3^- losses remained stable because the fertilization decrease was counteracted by a flow increase. Although many previous studies have predicted a future increase of N loads, our results showed that these ultimately depend on the future catchment land use and management. This is of utmost importance in Denmark where a reduction of N loads (70% coming from agriculture) is needed to meet the WFD objectives. The results of our study must, however, be interpreted with a certain caution due to a number of uncertainties. Especially, our analysis revealed that using different concentration pathways strongly influences nutrient load results.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2017.11.251.

Acknowledgements

This work was supported by the Danish Strategic Research Council as part of the IMAGE research alliance (IMAGE 2014, 09-067259) and by the EU Seventh Framework Programme, Theme 6 (Environment including Climate Change) as part of the MARS Project (Managing Aquatic ecosystems and water Resources under multiple Stress, contract No: 603378). Eugenio Molina-Navarro receives additional support from the Ramón Areces Foundation postgraduate studies programme. Authors also want to thank Dr. Karsten Bolding for his help with extracting and formatting climate change data and Anne Mette Poulsen for her valuable editorial comments.

References

Andersen, H.E., Kronvang, B., Larsen, S.E., Hoffmann, C.C., Jensen, T.S., Rasmussen, E.K., 2006. Climate-change impacts on hydrology and nutrients in a Danish lowland river basin. Sci. Total Environ. 365 (1–3), 223–237.

Arheimer, B., Lindström, G., 2015. Climate impact on floods: changes in high flows in Sweden in the past and the future (1911–2100). Hydrol. Earth Syst. Sci. 19, 771–784.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment part I: model development. J. Am. Water Resour. Assoc. 34, 73–89.

Arnold, J.G., Kiniri, R., Srinivasan, R., Williams, J.R., Haney, E.B., Neitsch, S.L., 2014. Soil & Water Assessment Tool. Input/Output Documentation. Version 2012. TR-439. Texas Water Resource Institute, College Station (TX).

Audet, J., Hoffmann, C.C., Jensen, H.S., 2011. Low nitrogen and phosphorus release from sediment deposited on a Danish restored floodplain. Ann. Limnol. 47, 231–238.

Bouraoui, F., Grizzetti, B., 2014. Modelling mitigation options to reduce diffuse nitrogen water pollution from agriculture. Sci. Total Environ. 468–469, 1267–1277.

Conley, D.J., Carstensen, J., Ærtebjerg, G., Christensen, P.B., Dalsgaard, T., Hansen, J.L.S., Josefson, A.B., 2007. Long-term changes and impacts of hypoxia in Danish coastal waters. Ecol. Appl. 17 (sp5), 165–184.

Daggupati, P., Yen, H., White, M.J., Srinivasan, R., Arnold, J.G., Keitzer, C.S., Sowa, S.P., 2015. Impact of model development, calibration and validation decisions on hydrological simulations in West Lake Erie Basin. Hydrol. Process 29, 5307–5320.

Devia, G.K., Ganasri, B.P., Dwarakish, G.S., 2015. A review on hydrological models. Aquat. Procedia 4, 1001–1007.

European Parliament and Council, 2000. Directive 2000/60/EC of 23 October 2000 establishing a framework for the Community action in the field of water policy. Off. J. Eur. Union 327, 1–73.

Faneca Sanchez, M., Duel, H., Alejos Sampedro, A., Rankinen, K., Holmberg, M., Prudhomme, C., et al., 2015. Report on the MARS scenarios of future changes in drivers and pressures with respect to Europe's water resources. Deliverable 2.1. Four Manuscripts on the Multiple Stressor Framework :pp. 215–284 Available at. http://www.mars-project.eu/index.php/deliverables.html, Accessed date: 26 April 2017.

Gassman, P.W., Reyes, M.R., Green, C.H., Arnold, J.G., 2007. The soil and water assessment tool: historical development, applications, and future research directions. Trans. ASABE 50, 1211–1250.

HELCOM, 2007. Baltic sea action plan. HELCOM Ministerial Meeting, Krakow.

Henriksen, H.J., Troldborg, L., Højberg, A.L., Refsgaard, J.C., 2008. Assessment of exploitable groundwater resources of Denmark by use of ensemble resource indicators and a numerical groundwater–surface water model. J. Hydrol. 348 (1–2), 224–240.

Hering, D., Carvalho, L., Argillier, C., Beklioglu, M., Borja, A., Cardoso, A.C., et al., 2015. Managing aquatic ecosystems and water resources under multiple stress — an introduction to the MARS project. Sci. Total Environ. 503–504, 10–21.

Hessen, D., 2013. Inorganic nitrogen deposition and its impacts on N:P-ratios and Lake productivity. Water 5 (2), 327–341.

Holsten, A., Vetter, T., Vohland, K., Krysanova, V., 2009. Impact of climate change on soil moisture dynamics in Brandenburg with a focus on nature conservation areas. Ecol. Model. 220, 2076–2087.

IPSL Climate Modelling Centre, 2017. CMIP5 Project. http://icmc.ipsl.fr/index.php/projects/100-international-projects/internation-projects-cmip5, Accessed date: 19 September 2017.

Jensen, N.H., Veihe, A., 2009. Modelling the effect of land use and climate change on the water balance and nitrate leaching in eastern Denmark. J. Land Use Sci. 4, 53–72.

Jeppesen, E., Kronvang, B., Meerhoff, M., Søndergaard, M., Hansen, K.M., Andersen, H.E., et al., 2009. Climate change effects on runoff, catchment phosphorus loading and lake ecological state, and potential adaptations. J. Environ. Qual. 38, 1930–1941.

Karlsson, I.B., Sonnenborg, T.O., Seaby, L.P., J, K.H., Refsgaard, J.C., 2015. Effect of a high-end CO2-emission scenario on hydrology. Clim. Res. 64, 39–54.

Karlsson, I.B., Sonnenborg, T.O., Refsgaard, J.C., Trolle, D., Børgesen, C.D., Olesen, J.E., et al., 2016. Combined effects of climate models, hydrological model structures and land use scenarios on hydrological impacts of climate change. J. Hydrol. 535, 301–317.

Kaspersen, B.S., Jacobsen, T.V., Butts, M.B., Boegh, E., Müller, H.G., Stutter, M., et al., 2016. Integrating climate change mitigation into river basin management planning for the water framework directive – a Danish case. Environ. Sci. Pol. 55 (1), 141–150.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World map of the Köppen-Geiger climate classification updated. Meteorol. Z. 15, 259–263.

Kovats, R.S., Valentini, R., Bouwer, L.M., Georgopoulou, E., Jacob, D., Martin, E., et al., 2014. Europe. In: Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, USA, pp. 1267–1326.

Kristensen, P., 2012. European Waters: Assessment of Status and Pressures. European Environmental Agency, Copenhagen.

Kronvang, B., Audet, J., Baattrup-Pedersen, A., Jensen, H.S., Larsen, S.E., 2012. Phosphorus load to surface water from bank erosion in a Danish lowland river basin. J. Environ. Qual. 41 (2), 304–313.

Lu, S., Kronvang, B., Audet, J., Trolle, D., Andersen, H.E., Thodsen, H., et al., 2015. Modelling sediment and total phosphorus export from a lowland catchment: comparing sediment routing methods. Hydrol. Process. 29, 280–294.

Miljø- og Fødevareministeriet, 2016a. Vandområdeplan 2015–2021 for Vandområdedistrikt Jylland og Fyn. Styrelsen for Vand- og Naturforvaltning, Copenhaguen.

Miljø- og Fødevareministeriet, 2016b. MiljøGIS. http://miljoegis.mim.dk, Accessed date: 26 April 2017.

Miljøministeriet, 2011. Vandplan 2010–2015. Odense Fjord. Hovedvandopland 1.13. Vanddistrikt: Jylland og Fyn. Miljøministeriet, Naturstyrelsen, Odense.

Molina-Navarro, E., Trolle, D., Martínez-Pérez, S., Sastre-Merlín, A., Jeppesen, E., 2014. Hydrological and water quality impact assessment of a Mediterranean limno-reservoir under climate change and land use management scenarios. J. Hydrol. 509, 354–366.

Molina-Navarro, E., Andersen, H.E., Nielsen, A., Thodsen, H., Trolle, D., 2017. The impact of the objective function in multi-site and multi-variable calibration of the SWAT model. Environ. Model. Softw. 93, 255–267.

Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., Veith, T.L., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASABE 50, 885–900.

Nash, J.E., Sutcliffe, J.V., 1970. River flow forecasting through conceptual models part I — a discussion of principles. J. Hydrol. 10, 282–290.
Neitsch, S.L., Arnold, J.G., Kiniri, R., Williams, J.R., 2011. Soil and water assessment tool. Theoretical Documentation Version 2009. Texas Water Resources Institute Technical Report No. 406. Texas A&M University System, College Station, Texas.
Olesen, J.E., Jeppesen, E., Porter, J.R., Børgesen, C.D., Trolle, D., Refsgaard, J.C., et al., 2014. Scenarier for fremtidens arealanvendelse i Danmark. 21. Vand og Jord, pp. 126–129.
Øygarden, L., Deelstra, J., Lagzdins, A., Bechmann, M., Greipsland, I., Kyllmar, K., et al., 2014. Climate change and the potential effects on runoff and nitrogen losses in the Nordic–Baltic region. Agric. Ecosyst. Environ. 198, 114–126.
Panagopoulos, Y., Makropoulos, C., Mimikou, M., 2011. Diffuse surface water pollution: driving factors for different geoclimatic regions. Water Resour. Manag. 25, 3635–3660.
Peng, J., Dan, L., Dong, W., 2014. Are there interactive effects of physiological and radiative forcing produced by increased CO2 concentration on changes of land hydrological cycle? Glob. Planet. Chang. 112, 64–78.
Refsgaard, J.C., Madsen, H., Andréassian, V., Arnbjerg-Nielsen, K., Davidson, T.A., Drews, M., et al., 2014. A framework for testing the ability of models to project climate change and its impacts. Clim. Chang. 122, 271–282.
van Roosmalen, L., Sonnenborg, T.O., Jensen, K.H., 2009. Impact of climate and land use change on the hydrology of a large-scale agricultural catchment. Water Resour. Res. 45, W00A15.
Schoumans, O.F., Silgram, M., Walvoort, D.J.J., Groenendijk, P., Bouraoui, F., Andersen, H.E., et al., 2009. Evaluation of the difference of eight model applications to assess diffuse annual nutrient losses from agricultural land. J. Environ. Monit. 11, 540–553.
Sellami, H., Benabdallah, S., La Jeunesse, I., Vanclooster, M., 2016. Quantifying hydrological responses of small Mediterranean catchments under climate change projections. Sci. Total Environ. 543 (Part B), 924–936.
Smed, P., 1982. Landskabskort over Danmark. Sheet 2–4. Copenhagen, Geografforlaget.
Statistics Denmark, 2014. Statistics Denmark. http://www.dst.dk/en, Accessed date: 26 April 2017.
Thodsen, H., 2007. The influence of climate change on stream flow in Danish rivers. J. Hydrol. 333, 226–238.
Thodsen, H., Hasholt, B., Kjærsgaard, J.H., 2008. The influence of climate change on suspended sediment transport in Danish rivers. Hydrol. Process. 22 (6), 764–774.
Thodsen, H., Baattrup-Pedersen, A., Andersen, H.E., Jensen, K.M.B., Andersen, P.M., Bolding, K., et al., 2014. Climate change effects on lowland stream flood regimes and riparian rich fen vegetation communities in Denmark. Hydrol. Sci. J. 1–15.
Thodsen, H., Andersen, H.E., Blicher-Mathiesen, G., Trolle, D., 2015. The combined effects of fertilizer reduction on high risk areas and increased fertilization on low risk areas, investigated using the SWAT model for a Danish catchment. Acta Agric. Scand. Sect. B Soil Plant 65, 217–227.
Thodsen, H., Windolf, J., Rasmussen, J., Bøgestrand, J., Larsen, S.E., Tornbjerg, H., et al., 2016. Vandløb 2015. NOVANA. Aarhus Universitet, DCE - Nationalt Center for Miljø og energy http://dce2.au.dk/pub/SR206.pdf, Accessed date: 19 May 2017.
Tian, F., Lü, Y.H., Fu, B.J., Yang, Y.H., Qiu, G., Zang, C., et al., 2016. Effects of ecological engineering on water balance under two different vegetation scenarios in the Qilian Mountain, northwestern China. J. Hydrol. Reg. Stud. 5, 324–335.
Trolle, D., Hamilton, D.P., Pilditch, C.A., Duggan, I.C., Jeppesen, E., 2011. Predicting the effects of climate change on trophic status of three morphologically varying lakes: implications for lake restoration and management. Environ. Model. Softw. 26, 354–370.
Trolle, D., Hamilton, D., Hipsey, M., Bolding, K., Bruggeman, J., Mooij, W., et al., 2012. A community-based framework for aquatic ecosystem models. Hydrobiologia 683, 25–34.
Trolle, D., Nielsen, A., Rolighed, J., Thodsen, H., Andersen, H.E., Karlsson, I.B., et al., 2015. Projecting the future ecological state of lakes in Denmark in a 6 degree warming scenario. Clim. Res. 64, 55–72.
White, K.L., Chaubey, I., 2005. Sensitivity analysis, calibration, and validations for a multisite and multivariable SWAT model. J. Am. Water Resour. Assoc. 41, 1077–1089.
Zhu, Q., Jiang, H., Peng, C., Liu, J., Fang, X., Wei, X., et al., 2012. Effects of future climate change, CO2 enrichment, and vegetation structure variation on hydrological processes in China. Glob. Planet. Chang. 80–81, 123–135.